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Diversity and Community Structure of Littoral Zone Macroinvertebrates in Southern Illinois Reclaimed Surface Mine Lakes

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ABSTRACT.—Surface mine lakes are common in parts of the Midwest, but little is known about the communities that inhabit them or the factors shaping them. Our objectives were to: (1) characterize littoral zone macroinvertebrate communities in surface mine lakes; (2) identify factors that influence macroinvertebrate diversity and community structure; and (3) examine the utility of some commonly used bioassessment metrics to assess habitat quality based on macroinvertebrates. Fourteen lakes on a 1133 ha parcel in southern Illinois were sampled in spring 2003 using a dip net that was swept over two (small lakes) or three (large lakes) littoral zone transects. Three hundred macroinvertebrates were randomly removed from each sample and identified. Stepwise multiple regressions were used to examine relationships between several physicochemical and biological variables and macroinvertebrates. Oligochaetes were typically the most abundant taxon, followed by *Hyalella*, Chironomidae, *Physella* and *Caenis*. Seventy percent of the macroinvertebrates collected from the lakes were predators, while collector-gatherers and scrapers constituted 20% and 9%, respectively. Macroinvertebrate richness was positively related to sunfish abundance and macrophyte cover ($R^2 = 0.91$, $P = 0.001$). Simpson diversity was positively related to lake area, percent rock and gravel substrates and simazine concentration and negatively related to bank slope and transparency ($R^2 = 0.92$, $P = 0.0003$). Richness and diversity metrics, % Oligochaeta, % Chironomidae, % insect taxa and % dominance varied across lakes, while a Hilsenhoff index and the EPT index appeared less useful for biological assessments in these habitats. Results indicate that macroinvertebrate communities in these lakes are typical of littoral habitats in other lentic systems and that macroinvertebrate diversity might be enhanced during creation and management of these systems by manipulating coarse mineral substrates and vegetation.

INTRODUCTION

Reclaimed surface mine lakes form when groundwater inundates a mining excavation. Thousands of these lakes were created in the Midwest during the first half of the 20th Century as a result of extensive coal mining (Castro and Moore, 2000). However, much of this mining ceased with the passage of the Surface Mining Control and Reclamation Act of 1977, which severely restricted the conditions in which mining was permissible. Additionally, in some regions like southern Illinois, much of the coal mined in this manner was sulfur-rich, and its use was restricted because of links with air pollution and acid precipitation.

Recent technological advancements have made the cleaner burning of sulfur-rich coal possible, and local economies that suffered from reduced mining have prompted efforts to implement these practices. For example, the southern Illinois coal mining industry will likely be rejuvenated in the near future and, as a result, the already numerous strip mine lakes in this region will soon become even more abundant. Given their abundance and potential ecological and economic significance throughout the region, understanding the diversity, structure and function of these systems is important.

Surface mine lakes can differ from natural lakes and reservoirs in a variety of ways. For one, unglaciated natural lakes found in the Midwest are usually shallower, having relative depths that are often less than 2% (Miller *et al.*, 1996; Doyle and Runnells, 1997; Castro and Moore, 2000). In contrast, deeper strip mine lakes may have relative depths approaching 40% (Miller *et al.*, 1996; Doyle and Runnells, 1997; Castro and Moore, 2000). This is important because the relation between depth and surface area is an important factor determining water circulation and the distribution of oxygen, nutrients and other important constituents (Anderson and Hawkes, 1985; Doyle and Runnells, 1997; Wetzel, 2001). Shallower natural systems are also more likely to experience seasonal turnovers that can facilitate mixing. In contrast, lakes that are very deep relative to their surface areas can remain stratified for long periods and, in some cases, even become meromictic (Hutchinson, 1957; Doyle and Runnels, 1997; Wetzel, 2001). Reservoirs, which are also abundant in the Midwest, are often more heterogeneous than strip mine lakes and they generally have at least seasonal inflows from associated streams. Inputs of sediments and other materials can increase turbidity and nutrient loads in reservoirs (Thornton *et al.*, 1991; Baxter, 1997; Rosenberg *et al.*, 2000). In contrast, strip mine lakes usually have low physical heterogeneity, no input from streams and effectively drain much smaller landscapes. Strip mine lakes generally have low turbidity, although, this, along with many other water quality parameters varies with adjacent land use.

Surface mine lakes have received some attention with regards to factors influencing water quality and recreational use, including studies on acid-leaching, general water chemistry and fish communities (*e.g.*, Miller *et al.*, 1996; Davis and Eary, 1997; Doyle and Runnells, 1997; Castro and Moore, 2000). However, there is little information regarding invertebrates, particularly in littoral zones where their abundance and diversity could be important to fisheries. Freshwater macroinvertebrates are also an important link between basal energy resources and higher order consumers in freshwater systems (*e.g.*, Merritt *et al.*, 1984; Healey, 1984; Boisclaire and Leggett, 1985; Wallace and Webster, 1996), and they can be effective indicators of system health and function (Gerritsen *et al.*, 1998; Barbour *et al.*, 1999).

Given the lack of information on invertebrate communities in strip mine lakes, our overall objective was to characterize littoral zone macroinvertebrate communities in fourteen southern Illinois strip mine lakes. These lakes are all in close proximity to each other, but vary in size, morphology and other physicochemical properties, as well as in fish communities. This allowed us to also examine a variety of physicochemical and biological factors that could potentially influence macroinvertebrate community composition. We also examined the potential utility of some commonly used macroinvertebrate-based bioassessment metrics for discriminating among lakes with different habitat quality.

METHODS

STUDY SITES

We studied 14 surface mine lakes located on 1133 ha in Randolph County, Illinois, that were recently purchased by the Illinois National Guard. This land, which was mined for coal and in which the excavations filled with groundwater ca. 20 y ago, is designated for future use as a National Guard training facility. This region of southern Illinois typically has a continental, seasonal climate and was historically a combination of mixed eastern deciduous forest with some tallgrass prairie. Exotic grasses now cover most of the research site, with restricted forested riparian areas (Fig. 1). Most land surrounding the property is in corn and soybean

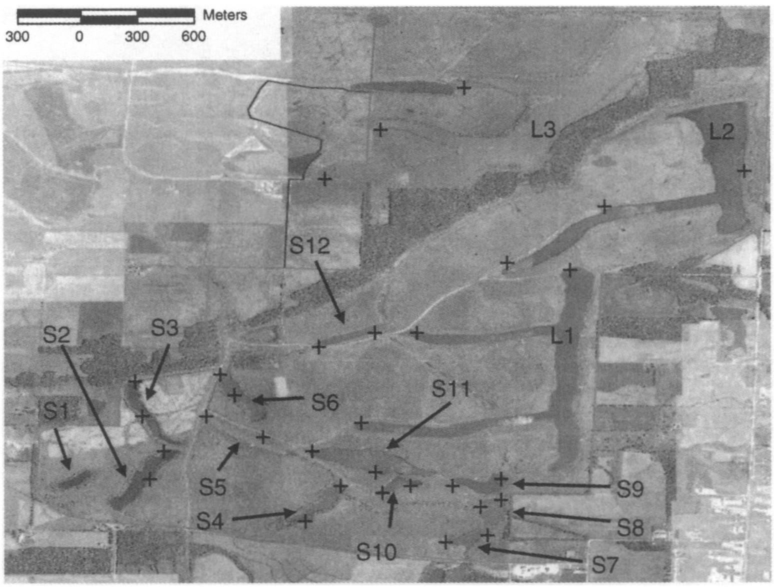

FIG. 1.—Satellite image of the fourteen study strip mine lakes in Randolph County, Illinois. Plus signs indicate macroinvertebrate sampling transects

row crops. Agrochemical concentrations within the lakes suggest that there is direct contact between the lakes and row crops or perhaps there may be groundwater contamination.

Of the 14 lakes examined, three (L1–L3) were considered large (36–56 ha) and the remaining eleven (S2–S12) were classified as small (1.4–8.8 ha) (Fig. 1; Table 1). The three large lakes have maximum depths from 25–29 m, and the maximum depth of the smaller lakes ranges from 3–11 m. Lake S1 was not considered during this study because it was inaccessible. Dissolved oxygen profiles indicate that these lakes are not permanently stratified (Garvey *et al.*, 2002), as is sometimes the case in strip mine lakes of the western United States (Miller *et al.*, 1996; Davis and Eary, 1997; Doyle and Runnells, 1997; Castro and Moore, 2000). As of April 2002 many of the smaller lakes had already stratified, with thermocline depths between 5–6 m (Garvey *et al.*, 2002). The study lakes also do not have acidity problems that characterize the copper and gold mining lakes of the western United States (Miller *et al.*, 1996) and the coal mining areas of the Appalachians (Ford, 1990). The pH in all study lakes ranged from 8.0–8.5 (Table 1) due to the high buffering capacity in this region and the addition of limestone blocks to further increase buffering capacity. All of the study lakes have limited littoral zones due to steeply sloping banks.

As is often the case with strip mine lakes, most of the study lakes were stocked with game fish, especially largemouth bass (*Micropterus salmoides*) and bluegill (*Lepomis macrochirus*). Other fish in these lakes, such as freshwater drum (*Aplodinotus grunniens*) and bowfin (*Amia calva*), were likely introduced from nearby Plum Creek during floods.

MACROINVERTEBRATES

We collected two macroinvertebrate samples in each of the smaller lakes (S2–S12) and three from the large lakes (L1–L3) during April–May 2003 (Fig. 1). We sampled during spring

TABLE 1.—Physicochemical features of the fourteen study strip mine lakes on the Sparta property in Randolph Co., IL, used for multiple regression analyses. L and S denote large lakes and small lakes respectively. All values are means for each lake. Maximum depth for lake L3 was not available

	L1	L2	L3	S2	S3	S4	S5	S6	S7	S8	S9	S10	S11	S12
Lake area (ha)	38.98	35.41	55.80	4.66	3.06	2.81	2.52	3.61	1.55	2.47	2.85	1.37	8.67	2.47
Max. depth (m)	28.74	31.21	N/A	10.17	7.09	9.15	3.22	11.37	5.00	8.17	12.94	9.32	12.69	6.77
Littoral Slope (Degrees)	9.09	8.70	16.70	14.44	11.29	4.58	11.03	7.07	7.55	19.03	10.30	24.82	13.90	12.79
Vegetation Cover (%)	93	68	33	98	38	08	80	05	28	18	63	100	08	03
Rocky Substrate (%)	0	0	0	0	10	10	40	40	0	0	10	0	10	20
pH	8.19	8.14	7.90	8.17	8.07	8.17	8.37	8.10	8.10	9.02	8.77	8.27	8.34	7.85
Conductivity (μS/ml)	0.61	0.72	0.32	0.31	0.60	1.28	0.81	0.82	0.87	0.69	0.54	0.96	0.57	1.84
Alkalinity (mg $CaCo_3$/liter)	105.89	142.28	80.44	110.00	148.08	205.71	161.08	160.33	162.50	141.08	132.71	224.17	112.08	253.17
Secchi Depth (cm)	168.06	115.01	183.81	236.50	57.92	38.06	48.00	61.33	57.20	59.14	74.00	86.78	78.76	54.00
Sunfish (#/hour)	325	96	461	0	205	55	191	134	9	114	33	0	231	31
Atrazine (μg/liter)	993.79	419.60	493.84	78.62	49.62	58.13	318.05	240.70	183.48	374.40	582.26	57.21	459.34	122.83
Simazine (μg/liter)	63.98	75.40	281.16	28.65	0.00	19.68	60.76	39.63	42.17	38.90	34.95	26.11	47.15	44.02

months because it is a period in this region when macroinvertebrate diversity is high and most insects are in later stages of development and, thus, larger, but have not yet emerged.

We selected individual sample sites in lakes that best represented the entire littoral zone conditions of the lake. For example, for a small lake that was sampled at two locations, we selected two different transects that best represented the prevailing shoreline and littoral habitat structures for that lake. Once a sampling site was chosen, a 1 m steel rod was driven into the substrate at the shoreline edge as a permanent marker. We used a 0.3 m × 0.5 m dip net (500 μm mesh) to make two parallel, non-overlapping sweeps along a 2 m transect perpendicular to the shore. Beginning 2 m out from the shore, the net was bumped along the substrate until the shore was reached to facilitate collection of shallow burrowing species as well as those in the vegetation and water column. Samples were rinsed in the net, placed in plastic bags and preserved in 8–10% formalin.

In the laboratory, 300 macroinvertebrates, or as many as were present if <300, were randomly removed from each sample using a gridded pan and random number table based on USEPA rapid biological assessment procedures (Barbour *et al.*, 1999). With the exception of Chironomidae and some non-insect taxa such as oligochaetes, we identified most to genus using Merritt and Cummins (1996) or Smith (2001). We based macroinvertebrate functional groups on primary designations in Merritt and Cummins (1996) and Smith (2001). Although oligochaetes were generally analyzed at the order level, the predacious *Chaetogaster* was abundant in our samples. Thus, we grouped *Chaetogaster* separately from other oligochaetes, which were categorized as collector-gatherers for functional analyses.

HABITAT ANALYSIS, WATER QUALITY AND FISH COMMUNITIES

We used a 2.0 × 2.0 m grid that was centered over each transect to estimate percent cover of submergent and emergent vegetation. Water depth was measured every 0.5 m along each transect to calculate slope and mean depth. A substrate grab sample was also collected every 0.5 m along each transect to estimate substrate composition based on a modified Wentworth scale (Cummins, 1962).

We used handheld meters to measure basic water chemistry parameters (pH, dissolved oxygen, temperature, conductivity) at various locations and depths within each lake during April of 2003 (Table 1). In addition, water column samples were also collected from all study lakes in April 2003 (depth transects at two locations in small lakes and three in large lakes) and analyzed for chlorophyll *a* and a variety of pesticides. Mean values for each lake from spring 2003 water chemistry and toxicology analyses were used in multiple regressions.

We sampled fish communities using catch per unit effort for 60 min of boat-mounted AC electrofishing. All fish were identified to species and all sizes and weights were recorded.

DATA ANALYSIS

We used macroinvertebrate data to calculate community metrics that reflect diversity (*e.g.*, direct count richness and Shannon and Simpson diversity), functional structure (*e.g.*, % functional groups) and tolerance [*e.g.*, Hilsenhoff index (HBI, Hilsenhoff, 1987,) EPT (Ephemeroptera, Plecoptera, Trichoptera)]. HBI values ranged from 0–10, with zero being the least tolerant and thus reflecting best conditions; individual tolerance values of taxa were based on Hilsenhoff (1987). Some uncommon taxa without published tolerance values were excluded from calculations of this metric. We calculated the EPT metric as the number of different taxa in the Ephemeroptera, Plecoptera or Trichoptera in a sample. We used a principal components analysis (PCA) to initially reduce the number of variables examined, and then used forward stepwise multiple regression to examine relationships between macroinvertebrate metrics and physicochemical and fish community variables (variables

TABLE 2.—Results of multiple regression analyses between environmental variables and macroinvertebrate diversity metrics. ± indicates positive/negative relationship

	R^2	P	P Leverage	Independent variable
Total Richness	0.91	0.0010	0.0007	+ Sunfish Abundance
			0.0010	+ % Vegetation
Dominance	0.63	0.0040	0.0027	– % Rock/Gravel
			0.0055	– Lake Area
Simpson Diversity	0.92	0.0003	<0.0001	+ Lake Area
			<0.0001	+ % Rock/Gravel
			0.0007	– Slope
			0.0019	+ Simazine
			0.0060	– Secchi
Shannon Diversity	0.60	0.0060	0.0041	+ % Rock/Gravel
			0.0077	+ Lake Area

listed in Table 1). All analyses were performed with JMP 14 version 4.0.3 (SAS Institute Inc., 2000).

RESULTS

MACROINVERTEBRATE COMMUNITIES

We identified 42 macroinvertebrate taxa representing 34 families, including 31 insects, four annelids, four mollusks and three crustaceans from the lakes. The number of taxa identified from an individual transect sample ranged from 8–19, with a mean of 11.6 ± 0.5 (mean ± 1 SE). Mean taxa richness in each lake ranged from a low of 8.5 in S7 to 15.3 in L3.

Non-insect groups dominated macroinvertebrate abundance. Oligochaetes accounted for 40% of total macroinvertebrates collected, and were the most dominant taxon in eight of the 14 lakes sampled. The predacious oligochaete, *Chaetogaster*, comprised ~70% of oligochaetes collected during this study. Lake S7, with 85% oligochaetes, was the most dominated by this group. *Hyalella* sp. was the second most abundant taxon, representing 22% of the total macroinvertebrates collected and was the most dominant taxon in lakes S8, S9 and S10.

Chironomidae was the most abundant insect taxon, and it was also the third most abundant taxon collected overall. Chironomids were the dominant taxon in three lakes (L3, S4, S6) and comprised 13% of the total macroinvertebrates collected during this study. Other common taxa included *Physa* (Gastropoda), *Caenis* (Ephemeroptera: Caenidae), *Enallagma* (Odonata: Coenagrionidae) and *Helisoma* (Gastropoda), each representing >2% of total macroinvertebrate abundance observed across all of the lakes.

Predators were the dominant functional group in the study lakes, representing, on average, 70% ± 1.8 of the total macroinvertebrates collected in each lake. The abundance of predators was consistent across all lakes; lake L2 had the lowest proportion of predators with 60% and Lake S4 had the highest proportion at 83%. Predator dominance was mainly attributable to *Chaetogaster*, while *Enallagma* and coleopterans (mostly Hydrophilidae larvae and Dytiscidae) also contributed.

Collector-gatherers, mostly chironomids and oligochaetes other than *Chaetogaster*, were the second most abundant functional group, averaging 20% ± 2.2 of the macroinvertebrates collected from each lake. Scrapers (mostly *Physa* and *Caenis*) were the next most abundant group, averaging 9% ± 1.7 of the total macroinvertebrates collected from each lake. Average scraper percentages within lakes ranged from 1% in S3 to 22% in S8.

MACROINVERTEBRATE COMMUNITY METRICS AND PHYSICOCHEMICAL VARIABLES

Multiple regression indicated that taxa richness was positively related to abundance of sunfish and the presence of vegetation ($R^2 = 0.91$, $P = 0.001$, Table 2). Percent dominance ranged from 26% in Lakes L1 and S6 to 85% in Lake S7 and was negatively related to lake area and % rocky substrates ($R^2 = 0.63$, $P = 0.004$, Table 2). Shannon diversity ranged from 0.67 in S7 to 1.99 in L3, with a mean of 1.4 ± 0.6 across lakes, and was positively related to percentage of rocky substrate and lake area ($R^2 = 0.6$, $P = 0.006$, Table 2). Simpson diversity ranged from 0.17 in S6 to 0.71 in S7, with a mean of 0.36 ± 0.03, and was positively related to lake area, % rocky substrate and simazine concentration, and negatively related to slope and Secchi depth (Table 2). Overall, Simpson diversity showed a stronger relationship with lake variables than Shannon diversity.

Evenness (0.27 ± 0.05), % Oligochaeta (47 ± 3%), % Chironomidae (36 ± 4%) and % insect taxa (29 ± 5%) metrics varied across lakes, but were not significantly related to any of the variables we examined. The HBI (8.0 ± 0.06) showed little variability, failed to discriminate among lakes and indicated all were in poor condition. The EPT taxa metric (2.0 ± 0.2) was also of little use for discriminating among lakes because there were very few taxa from these orders collected.

DISCUSSION

MACROINVERTEBRATE COMMUNITIES

This study is one of the few that we know of to examine littoral zone macroinvertebrate communities in strip mine lakes, despite their ubiquity in some areas. We found that the littoral zones of these lakes were dominated by only three taxa, but high abundances of oligochaetes and chironomids is typical of many freshwater systems. For example, oligochaetes and chironomids are often dominant in the littoral zones of other types of lakes in many regions (Mittelbach, 1981; Beckett *et al.*, 1990; Dougherty and Morgan, 1991; Weatherhead and James, 2001), as well as in some prairie wetlands (Zimmer *et al.*, 2001) and higher order streams and rivers (Barton, 1980; Quinn and Hickey, 1990). The wide distribution and abundance of oligochaetes and certain chironomid taxa in freshwater systems is often attributed to their ability to thrive in fine substrates that are not suitable habitat for many other groups (Weatherhead and James, 2001; Barton, 1980). Fine substrates such as silt and sand are relatively unstable and can be oxygen limited just below the sediment-water interface because of a lack of interstitial spaces and mixing (Allan, 1995). Fine substrates often dominate low gradient streams, eutrophic lakes and reservoirs, and wetlands, and were the predominant substrates in the littoral zones of these strip mine lakes.

The amphipod *Hyalella* was abundant in most of the lakes we examined, particularly in lakes L1, S5, S8, S9, S10 and S11, where *Hyalella* accounted for >15% of the macroinvertebrates collected. *Hyalella* is the most widely distributed amphipod in North American freshwater systems (Bousfield, 1958). It is omnivorous, frequently grazing on periphyton and detritus (Cooper, 1965; Hargrave, 1970) and has typically been found in abundance among rooted vegetation and coarse substrates within littoral zones of lentic systems (Strong, 1972; Pennak and Rosine, 1976). Additionally, Strong (1972) observed that *Hyalella* was very effective at using thick algal growth for cover in Oregon coastal lakes. The six lakes in which we found the highest proportions of *Hyalella* did not have more vegetation or coarse substrates than the other lakes, but they did have abundant, thick algal mats along the shoreline, and it is possible that *Hyalella* were attracted to the additional structure that these mats provided.

Simazine, a commonly used herbicide, was present in most of these lakes and its concentration was one of five variables related to Simpson diversity in multiple regressions.

We cannot identify the underlying mechanisms for this relationship with this descriptive study. However, simazine has often been used to control algae and can alter aquatic communities at concentrations less than those we observed in the lakes (Strandberg and Scott-Fordsmand, 2002) and it is possible that the positive relationship with Simpson diversity was related to reductions or community composition changes in algae and/or differential sensitivities of invertebrate taxa.

Atrazine, another herbicide commonly used in the Midwest, was at unusually high concentrations [307 ± 71.0 μg/liter (mean ± 1 SE)] in all of the study lakes. Atrazine concentrations as low as 1 μg/liter can inhibit photosynthesis in some algal species (deNoyelles *et al.*, 1982; Herman *et al.*, 1986; DeLorenzo *et al.*, 2001), while 20 μg/liter is sufficient to alter phytoplankton community composition (deNoyelles *et al.*, 1982). Dewey (1986) examined the effects of different concentrations of atrazine on aquatic insects in mesocosms and found that both total richness and abundance were significantly reduced at 20 μg/liter and that both decreased further with increasing concentrations. Dewey (1986) also found that predatory insects appeared to be the least affected by atrazine, presumably because of food quantity and/or quality changes for primary consumers. Given that we observed unusually high concentrations, it is possible that the high proportion of predators we observed was linked at least in part to atrazine. The lack of statistical relationships between macroinvertebrates and atrazine across the lakes we examined was likely due to its abundance in all lakes, and thus a lack of an adequate gradient of concentrations to detect patterns. Additionally, invertebrate communities can integrate conditions over time and the concentrations we measured in spring 2003 may not be representative of concentrations in each lake over longer time periods.

MACROINVERTEBRATE COMMUNITY PATTERNS ACROSS LAKES

Macroinvertebrate taxa richness was highest in lakes with higher macrophyte cover and sunfish abundance and the mechanisms behind these patterns are likely complex. Other studies indicate that freshwater invertebrate abundance and diversity is often positively related to macrophyte diversity and density and this has been attributed to increased structural heterogeneity, refugia from predation and food resources (*e.g.*, Mittelbach, 1981; Hanson, 1990; Beckett *et al.*, 1990; Batzer and Wissinger, 1996). Further, different types of macrophytes can support different invertebrate communities (Hanson, 1990; Jeffries, 1993). Eurasian milfoil (*Myriophyllum spicatum*) and near-shore emergent sedges, reeds and grasses dominated the macrophyte communities in the littoral zones of the lakes we studied.

Simpson diversity was negatively related to bank slope and this relationship was likely also linked to relationships with macrophytes, as the steepness of the littoral zone in lakes can limit macrophyte abundance. As an example, Duarte and Kalff (1986) found that in a north temperate lake bordering Quebec and Vermont, macrophyte biomass decreased as littoral zone slope increased. They suggested this relationship was likely a result of the erosional nature of steep slopes that causes much of the organic sediments to move away from the littoral zone to deeper areas. Gentle slopes, on the other hand, are better for retaining fine organic sediments and provide more stability. Additionally, gently sloping littoral zones have increased surface area in the photic zone.

Sunfishes are known to feed on littoral zone macroinvertebrates, but whether they regulate prey communities is still debated. Many studies indicate that fish effects are often minimal because of invertebrate defense mechanisms and/or lower predator efficiency in macrophytes (*e.g.*, Mittelbach, 1981; Crowder and Cooper, 1982; Gilinsky, 1984; Hershey, 1985; Pierce and Hinrichs, 1997; Zimmer *et al.*, 2001). In fact, fish predation can sometimes lead to an increase in the abundance of some invertebrates, particularly when certain types or sizes of prey are selectively taken, ostensibly reducing competition and predation pressures for others (Mittelbach, 1981; Morin, 1984). It is possible that the positive relationship that we observed between macroinvertebrate richness and sunfish abundance was a result of sunfish influencing macroinvertebrates through these mechanisms. Likewise, both sunfish and macroinvertebrates may simply have been responding positively to increased density of macrophytes in some of the lakes.

We found that larger lakes with more large, rocky substrates were more diverse and, thus, less dominated by one or a few taxa. This is likely related to increased habitat heterogeneity associated with both larger habitat areas and larger substrate particle sizes. Schmude *et al.* (1998) compared the stream invertebrate communities of complex, three-dimensional artificial substrates to simpler two-dimensional substrates and found significantly higher abundances and richness in the complex substrates. They attributed these differences to greater heterogeneity, surface complexity, interstitial spaces and surface area of the complex substrates. These factors, along with the increased stability associated with larger substrates, are likely reasons for higher macroinvertebrate diversity that we observed in rock and gravel substrates in these lakes.

Of the 15 bioassessment metrics that we examined during this study, richness, % dominance, Shannon diversity and Simpson diversity all showed significant relationships with physicochemical variables in multiple regression. Additionally, evenness, % Oligochaeta, % Chironomidae and % insect taxa varied across lakes and may, thus, be useful for assessments, although they did not show significant relationships with the variables we used in regression models. Although used extensively in stream assessments, the Hilsenhoff Biotic Index (HBI) and EPT index showed little variability and indicated poor conditions in all lakes because the invertebrate communities in these systems were composed primarily of tolerant taxa.

Our results indicate that the macroinvertebrate communities in the littoral zones of these mine lakes are primarily limited by physical habitat quality, including mineral substrates, bank slope and vegetation and that pesticides may also be influencing them. As such, management of existing systems and construction of new ones should consider: (1) increasing habitat heterogeneity and stability by increasing coarse, rock substrates, reducing littoral zone slopes and enhancing macrophytes in littoral zones; and (2) reducing agrochemical inputs through planting of riparian buffers and controlling inputs on adjacent fields. Understanding how to enhance macroinvertebrate communities and overall habitat quality in strip mine lakes is important, as most of the thousands of existing systems are managed for recreational fishing and the current revitalization of coal mining in some regions means they will become an increasingly abundant freshwater resource.

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